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LCA OF WASTE MANAGEMENT SYSTEMS

Reliability-based life cycle assessment for future solid waste management alternatives in Portugal

Ana Pires · Ni-Bin Chang · Graça Martinho

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Abstract
Background, aim, and scope This paper presents a study related to the application of the reliability-based life cycle assessment (LCA) to assess different alternatives for solid waste management in the Setúbal peninsula, Portugal. The current system includes waste collection, transport, sorting, recycling, and mechanical and biological treatment (MBT) by means of aerobic treatment and landfill. In addition, some future expansion plans are discussed.
Materials and methods The proposed 18 alternatives were examined with respect to six impact categories based on a customized life cycle inventory (LCI). All the alternatives are designed to comply with the targets prescribed in the Packaging and Packaging Waste Directive and the Landfill Directive. These 18 alternatives were eventually assessed by using the reliability-based LCA methodology with respect to some uncertain parameters and scenarios.
Results and discussion The results show that solutions based on anaerobic digestion at the MBT followed by energy recovery are the most advantageous options. Overall, recycling may help to avoid most environmental impacts. Alternatives which treat massively biodegradable municipal waste are also competitive. In addition to the recycling options, electricity production is also an influential determinant which affects the results. The uncertainty analysis focused on testing different energy-from-waste options (like landfill and MBT biogas electricity production) and different recycling substitution ratios. Such a quantitative analysis is proved effective to confirm the reliability of the LCI in the study.
Conclusions In order to improve the sustainability of the solid waste management (SWM) system, final suggestions may concentrate on the closure of aerobic MBT, the enhancement of anaerobic digestion MBT treatment, and the maximization of energy recovery from high calorific fractions of the waste streams. However, the option of stabilized residue applications cannot be encouraged at this stage, especially due to the absence of Portuguese regulations to control the quality of organic products issuing from biological treatment units.

Keywords LCA · MBT · Municipal solid waste management system · RDF · Uncertainty analysis

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A. Pires (✉) · G. Martinho
Departamento de Ciências e Engenharia do Ambiente, Faculdade de Ciências e Tecnologia, Universidade Nova de Lisboa, 2829-516, Caparica, Portugal
e-mail: alp11931@fct.unl.pt

N.-B. Chang
Department of Civil, Environmental, and Construction Engineering, University of Central Florida, 4000 Central Florida Blvd., Orlando, FL 32816, USA

1 Introduction

Life cycle assessment (LCA) is an important tool complementing other systems analyses for sustainable development in an urban region. To achieve the sustainability goals, however, the long-established cost-effectiveness approach is becoming obsolete whereas cost–benefit analysis may also have to be revised in the future. It is believed that LCA-based planning techniques in concert with cost–benefit information may lead to an insightful assessment of sustainable solutions for solid waste management (SWM).

Applications of LCA techniques to SWM systems in Europe started in the 1990s of the last century. The first case of its kind took place for assessing the management of waste beverage packaging systems in Denmark. This pioneering work aimed to determine the best solutions for

different packaging waste materials such as paper, cardboard, and glass (Dalager et al. 1995; Ekvall et al. 1998; Frees et al. 1998; Frees and Weidema 1998; Person et al. 1998a, b; Ryberg et al. 1998). The evaluation of the existing technologies to treat biodegradable municipal waste (BMW) was also carried out (Dalemo et al. 1997). Since the beginning of the twenty-first century, the life cycle inventory (LCI) and the LCA applications have been widely applied to many real world cases due to the fast growth of software packages exclusively for SWM. These include but are not limited to IWM-1 and 2 (White et al. 1995; McDougall et al. 2001), WISARD/WRATE (Ecobilan 2004; Buttol et al. 2007), and EASEWASTE (Christensen et al. 2007). In parallel with the rapid development of such tools, the general-purpose LCA software packages such as SimaPro (Pré Consultants), GaBi (PE International), CMLCA (Leiden University), and TEAM (Ecobilan) have also been customized to conduct a wealth of LCA practices for waste management (Ekvall and Finnveden 2000; Finnveden et al. 2002; Moberg et al. 2002; Muñoz et al. 2004; Finnveden et al. 2005).

Kirkeby et al. (2005) assessed technology-based SWM scenarios for the Aarhus municipality in Denmark and the LCA confirmed that there is no significant difference between anaerobic digestion and incineration based on their LCI. Using a Spanish municipality as a test case, Bovea and Powell (2006) found that the SWM planning scenarios with energy recovery may achieve significant improvements in terms of mitigation of environmental impacts. Rodríguez-Iglesias et al. (2007) showed that incineration without pre-treatment of waste streams is the worst scenario, and proper integration between anaerobic digestion and incineration would certainly lead to a better option. Escalante et al. (2007) and Buttol et al. (2007) pointed out that recycling with energy recovery is one of the most advantagous options from the environmental point of view. Recently, de Feo and Malvano (2009) discussed the assessment of 12 scenarios showing that a SWM system based on recycling and material recovery without incineration would be preferable. Some studies like that of Tunesi and Rydin (2009) have not reached any final conclusion about which would be the best way to manage municipal solid waste (MSW).

A previous review showed that LCA results for SWM systems do not reach the same conclusion oftentimes, even following the ISO Standards family—ISO 14040 (ISO 2006a), which ensures minimization of assessment discrepancies based on the same set of criteria. This uncertainty may be due to: (1) the methodological assumptions being made are uncertain and may potentially influence the results (Finnveden et al. 2009); (2) the system boundaries considered by these studies, in regard to whether or not to include specific equipment or life cycle emissions of energy consumed by SWM systems may lead to imprecise conclusions (Cleary 2009); (3) the data used for LCA, which translate geographic differences between the data sources and the location of the study, could also make the conclusion biased (Cleary 2009); (4) the lack of data for validation of the LCI applied may end up as increased uncertainty in LCA (Winkler and Bilitewski 2007) and can restrict the conclusions that may be taken from a specific study (Finnveden et al. 2009); and (5) the impact categories assessed, due to the fact that different coverages can affect the conclusions because not all types of category impacts are equally well understood in a typical LCA (Finnveden et al. 2009). Hence, a typical LCA with a variety of conditions may compound state-of-the-art data analytical techniques which would be of importance in the LCA-based decision analysis.

In Portugal, the LCA applications for SWM decision making have been rarely applied. The only application found in the literature is the LCA for the Oporto municipality conducted by Xará et al. (2005). Nevertheless, it is vital to ensure the compliance of SWM according to the European Waste Management Directives, like the Packaging and Packaging Waste Directive 2004/12/EC (EC 2004) and the Landfill Directive 1999/31/EC (EC 1999), both of which brought environmental perspectives into economic decision making. Besides, the New Waste Framework Directive 2008/98/EC (EC 2008) brought new perspectives into waste management based on waste hierarchy. Such hierarchical preference for waste management implies the desirable priority of reduction, reuse, recycling, energy recovery, and disposal in sequence. However, when applying the referred waste hierarchy, Member States in European Union (EU) will be expected to take measures to encourage the options which deliver the best overall environmental outcome. This may require specific waste streams departing from the hierarchy, and justifying life cycle implications with "system thinking" in association with the overall impacts of the production and management of such types of waste (EC 2008). During the field implementation, site-specific LCA at the local level is essential for the characterization of an all-inclusive or most relevant impact assessment since the LCA outcome being conducted and acquired for other places might not always be transferable.

This paper uniquely develops an LCA to analyze the environmental impacts produced by a SWM system in Portugal. It takes 18 specific management alternatives into account in a comparative way individually or collectively. These alternatives cover several combinations of biological treatments with or without source separation of BMW. All of the alternatives consider a common issue of packaging waste collection to reach packaging waste recycling targets in the prescribed Directive. The inclusion of LCA in waste management leads to improvement of the understanding of how a SWM system can comply with the legislative requirements and search for the optimal alternatives from environmental and social perspectives simultaneously. In

this paper, based on the ISO 14040 Standards, such an LCA was carried out and followed by an uncertainty analysis along with a reliability assessment.

2 Description of the study area

The Setúbal peninsula is located in the district of Setúbal with an area of 1,522 km^2 and has 714,589 inhabitants (AMARSUL 2009). The area is divided in nine municipalities, as shown in Fig. 1. With a regionalization basis, AMARSUL is a company owned by the local municipalities, which has been made responsible for managing the MSW since 1997. This SWM system is composed of nine recycling centers, two material recovery facilities (MRFs), two landfills, one transfer station, and one aerobic mechanical biological treatment (MBT).

Nowadays, AMARSUL promotes the separation of paper/cardboard, glass, and light packaging (plastics, metals, and composite packaging) waste by means of curbside recycling systems. Each type of waste is collected separately in three specific containers, and then sent directly to the MRF for recycling, material recovery, and reuse. The remaining waste fractions in households, which are normally destined for final disposal at landfills, are then collected through a door-to-door and/or bin collection scheme. In the case of Sesimbra municipality, the waste stream is first sent to the transfer station, and then finally disposed of at sanitary landfills. The residual waste after waste separation and recycling collected from Setúbal municipality is transported to an aerobic MBT plant where the "stabilized residue" can be converted as fertilizer to be applied as agriculture soil-amendment materials.

Within this MSW system, it has recently become necessary to make some changes in order to comply with the Packaging and Packaging Waste Directive (EC 2004) and Landfill Directive (EC 1999). The National Plan for MSW (designated as PERSU II) decided to pursue the construction of several more MBT units. An anaerobic digestion (AD) MBT unit, with a mechanical treatment to separate recyclables and high calorific material to produce refuse-derived fuel (RDF), is under planning. It is expected that this unit will work with two separate lines, one of which is related to the biodegradable municipal solid wastes (BMW) and the other is for the residual waste streams. The RDF may be combusted in an incinerator to generate electricity. The existing aerobic MBT plant will be maintained as usual. It is expected that both MRF plants, which are currently fitted with manual sorting, will be later fitted with two automatic sorting units.

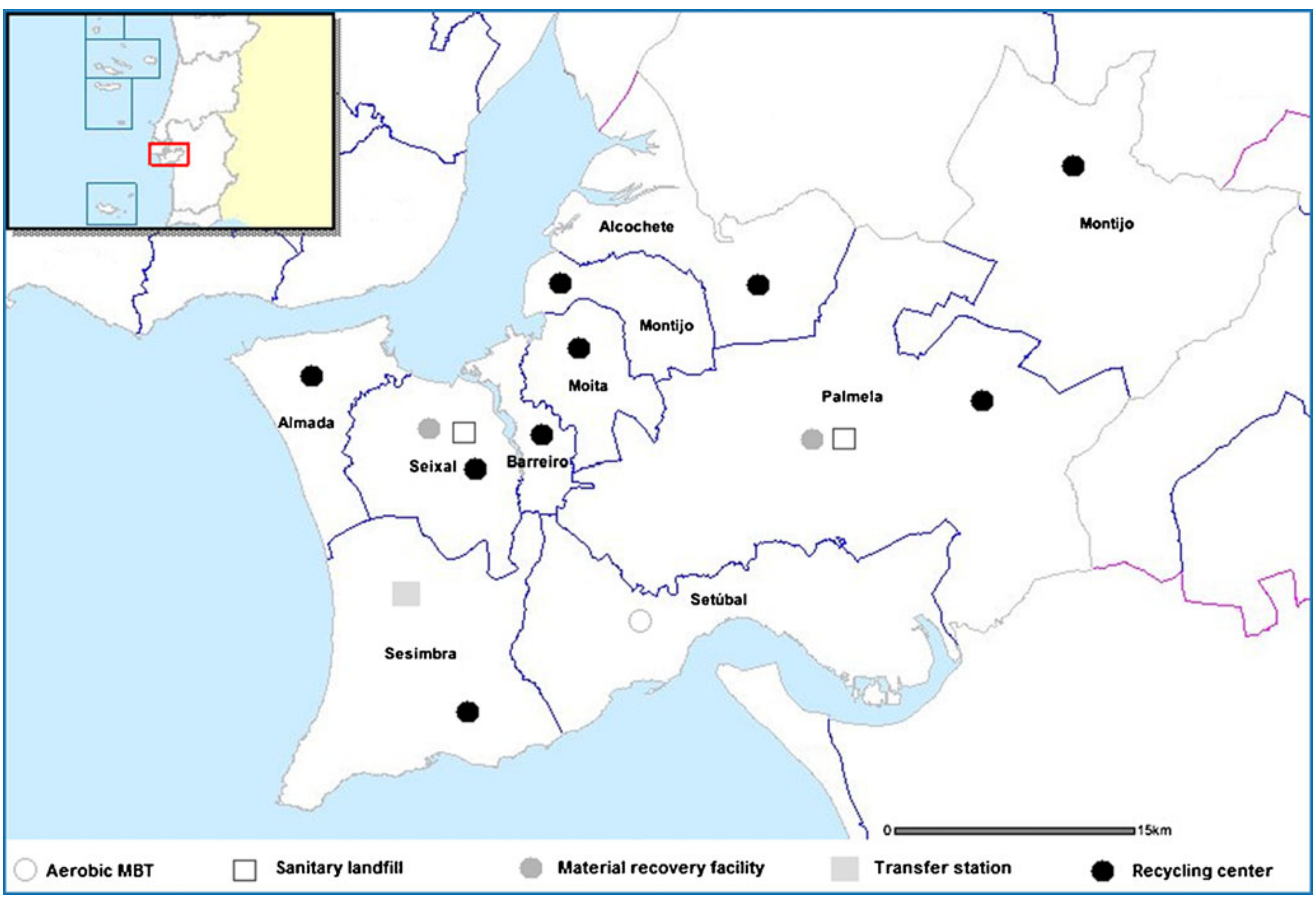

Fig. 1 The geographical location of the Setúbal peninsula SWM system

3 Materials and methods

In this paper, a customized LCA methodology was developed and applied to conduct a comparison of waste management alternatives for the Setúbal peninsula SWM system. According to the ISO 14040 (ISO 2006a), a LCA consists of four major stages: goal and scope definition, life cycle inventory, life cycle analysis, and interpretation of the results. The following sections present a detailed description of each stage in our application.

3.1 Goal and scope definition

The aim of the study was to apply the LCA procedure to the SWM system of the Setúbal peninsula in order to compare waste management alternatives subject to the targets associated with both the Packaging Waste Directive and the Landfill Directive in such a way that could promote sustainable development. A schematic of the SWM to be analyzed is shown in Fig. 2, which generally covers all stages of SWM involved from raw waste pick-up to the delivery to bins, to some intermediate processing units, and to the final disposal at landfills. Both anaerobic digestion MBT lines are represented as two separate processes with and without RDF production. The LCA provided in this paper is of attributional type. We applied the “zero burden assumption”, suggesting that waste carries none of the upstream environmental burdens into the SWM system (Ekvall et al. 2007).

These SWM processes include collection and transportation of residual waste and recyclables, waste treatment, waste transport from waste treatment facilities to the final destination, energy-from-waste or waste-to-energy, and landfilling. Several final destinations for recyclables are located in Spain rather than Portugal, specifically for the cases when handling composite packaging and ferrous and non-ferrous metals packaging materials.

Based on this system, Table 1 presents the 18 management alternatives for assessment plus the current situation (base scenario). These alternatives include waste collection and separate recycling of the three packaging materials through bin systems, which handle 12.4% of the current MSW in the study area. This MRF system is responsible for compliance with the prescribed target in the Packaging Waste Directive.

Alternative 0 refers to the predicted change that will take place in the Setúbal peninsula waste management system. The remaining alternatives were designed to examine some special options for complying with the Landfill Directive. For example, alternative 1 emphasizes the inclusion of aerobic MBT; alternative 4 signifies the use of AD MBT; alternative 6 examines the specific case of using a BMW anaerobic digestion line. In general, alternatives 0, 3, and 5 are options for differing intermediate processing. Separation of high calorific fractions of waste for energy recovery was considered through the production of RDF and the direct burning of high calorific fractions in municipal incinerators.

The creation of Table 1 is based on the total amount of waste produced in 2008, which is 421,726 tonne. According to Finnveden (1999), having identical amounts of waste treated in different scenarios makes it possible to simplify a comparative analysis by neglecting the production and use of the materials. Based on the investigation of average waste composition data of MSW region wide, the waste stream has 31.69% putrescibles, 14.13% paper and cardboard, 11.35% of plastics, 5.83% of glass, 4.14% of composites, 1.82% of metals, 2.07% of wood, 11.72% of textiles, 15.33% of fine particles, and 1.92% of others (EGF 2009, personal communication, Empresa Geral do Fomento). Hardware equipment, such as bins, buildings, and trucks, were excluded from the LCA. However, the use of fuels, electricity, and auxiliary materials for shipping and handling were included in the LCA. The Portuguese electricity generation mix considered is composed of 28.1% of coal,

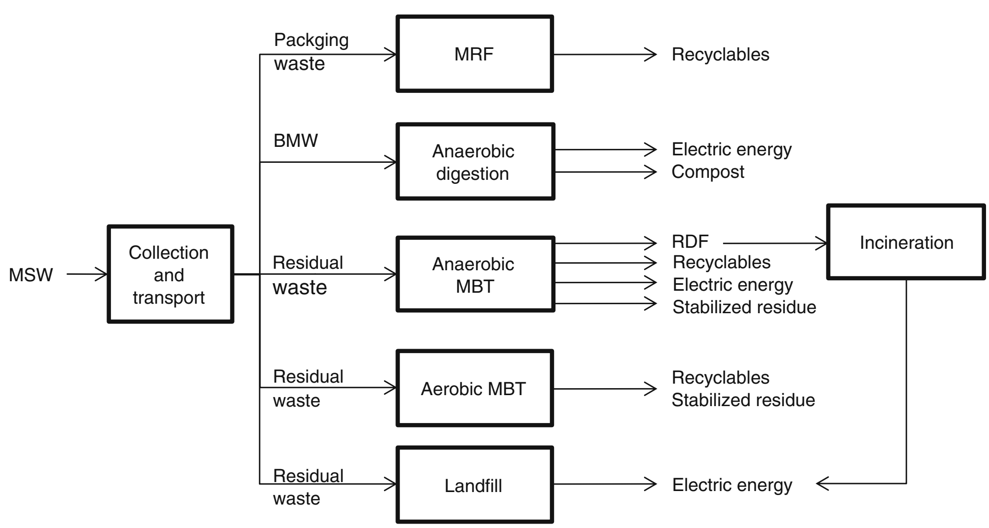

Fig. 2 Schematic of the SWM system at Setúbal Peninsula

Table 1 The distribution of waste streams associated with each alternative in the SWM system

Fraction (%) option	Alternatives							
	0/0[a]/0[b]	1/1[a]	2/2[a]/2[b]	3/3[a]	4/4[a]/4[b]	5/5[a]/5[b]	6/6[a]	Base
MRF	12.4	12.4	12.4	12.4	12.4	12.4	12.4	4.8
Anaerobic digestion BMW	5.4	0	0	13.3	0	7.5	28.7	0
Anaerobic digestion MBT	28.2	0	33.9	0	49.6	38.9	0	0
Aerobic MBT	13.2	49.7	15.8	32.6	0	0	0	13.8
Landfill with ER	40.8	37.9	37.9	41.7	38.0	41.2	58.9	81.4

[a] Alternatives considering RDF production plus incineration of high calorific fraction

[b] Alternatives not considering RDF production but incineration of high calorific fraction

8.37% of fuel oil, 30.5% of natural gas, 0.55% biomass, hydro 25%, waste 7%, geothermic 0.33%, and wind 0.15%.

In an LCA with multiple products, as in our study, it is necessary to set up the methodological framework. According to the ISO 14044 (2006), the system boundary should be geared toward expanding the product system to include the additional functions related to the co-products to avoid allocation. In this LCA, the material recycling, energy recovery, and fertilizers application (i.e., stabilized residue waste) of MSW were included in the LCA as co-products, which collectively resulted in an expansion of the system boundary. In this LCA, the emissions resulting from the referred operations were included as the baseline information as the emissions of those competing products and energy recovery potential resulting from those alternative operations were also considered for the purpose of comparisons. In this context, the system can be expanded to include additional burdens of co-product processing and the avoided burdens of any processes being dropped (Tillman et al. 1994; Guinée et al. 2002; Thomassen et al. 2008; Finnveden et al. 2009).

To ensure a correct implementation in regard to the avoided burden through successful MSW recycling and reuse, the co-products in the expanded system boundary should have the same function as the raw products. The substitution ratios are then applied considering closed-loop and open-loop procedures. Table 2 presents the substitution ratios for recovered materials and energy consumed. In the cases where the substitution ratio is 1:1, they have been considered as a closed-loop procedure, allowing the hypothesis that no changes occur in the inherent properties of the recycled materials (Rigamonti et al. 2009b). For example, 1 kg of recycled glass can replace 1 kg of virgin glass without considering degradation of the material during the recycling so that the quality of the secondary material may not be worse than that of the primary material (Rigamonti et al. 2009b). The materials included in this situation are glass, metals, polyethylene (PE) plastics, plastic wood, fertilizers, and electricity. Specifically, 15% of the electricity consumed in Portugal was purchased from Spain, and the ratio can also be taken into account, with a proportion of 85/15, for carrying out the LCA. PE, expandable polystyrene (EPS), and plastic wood are specific cases having 1:1 substitution ratio, since they only occur once in the sense that degradation of the material is not considered.

In the cases where the substitution ratio is <1, an open-loop allocation procedure is applied since degradation of the material should be considered, like the cases of polyethylene terephthalate (PET), paper/cardboard, and paper from composite packaging. The calculation of substitution ratios was based on the limited number of times through which a specific material can be recycled and reused repeatedly (Rigamonti et al. 2009a). For PET, the limit number of recycling with respect to losing physical properties consid-

Table 2 Products obtained from the SWM system and the assumptions for LCA

Product obtained	Substitutes assumed	Substitution ratio assumed
Cardboard from recovered paper and cardboard	Cardboard from virgin pulp	1:0.833
Glass produced from recovered glass processed	Glass from virgin materials	1:1
Tubes from PE recycled	Tubes from virgin PE	1:1
Multi-layer packaging materials from recycled PET	Multi-layer packaging from virgin PET	1:0.625
Recycled EPS lightweight soil	Virgin EPS lightweight soil	1:1
Paper from composite packaging materials recycled	Paper from virgin pulp	1:0.625
Outside furniture blocks from recycled mixed plastics	Outside furniture blocks from wood	1:1
Ferrous metals from recycled ferrous metals	Pig iron	1:1
Aluminum ingot from recycled aluminum metals	Aluminum ingot from virgin aluminum	1:1
Compost	N, P, K, Ca, and Mg fertilizers	1:1 (based on nutrient content)
Electricity	Electricity mix consumed in Portugal	1:1

ered was five times (Comieco 2008). Concerning paper from composite packaging, the same limit may be applied given that the proportion of paper in those packaging (0.75%) may be assumed and the calculation procedure adopted by Rigamonti et al. (2009b) may be applied. The substitution ratio adopted for PET is from the Institute for Prospective Technological Studies (Delgado et al. 2007).

3.2 Life cycle inventory

The life cycle inventory is the second phase of the LCA. It is an inventory of input/output data related to the SWM system that is being studied. It involves the collection of the data which is necessary to meet the goals of the defined study (ISO 2006b). In accordance with the scope of the study, an LCI was prepared for the waste management activities specified in Fig. 2. The Umberto 5.5 software package was used to support the LCA.

Concerning each operational unit analyzed in the AMARSUL system, a short description of the data and assumptions considered for prescribed scenarios is provided. First of all, some of the information applied for our systems analysis was provided by the Empresa Geral do Fomento (EGF 2009, personal communication, Empresa Geral do Fomento), co-owner of the SWM system at the AMARSUL, which is responsible for the management of this MSW system, and the Portuguese Environment Agency (APA). The rest of information was supplied by the Umberto software library and the selected data sources such as machinery specifications provided by the vendors.

3.2.1 Waste collection and transport

Waste collection is routinely performed by municipalities. The service can be carried out by municipalities or by hiring private collection companies. MSW is temporary discarded into road-side containers (bins), and collection vehicles can remove waste inside the bins periodically. Table 3 lists the requirements needed to perform the collection and transport processes. Transportation between those operational units inside the AMARSUL system was not considered since the AMARSUL does not have a BMW collection system. The approach used herein was to assign the same shipping distance and diesel fuel consumption to the municipalities that may treat BMW in a future AD MBT unit in parallel.

In the AMARSUL system, the MSW composition is as follows: 70% of food waste, 15% of green waste, 5% of plastics, 1.9% glass, 0.25% ferrous metals, 0.15% of non-ferrous, 0.65% of others, and 7.05% of fines, adapted from a BMW characterization program of Lisbon metropolitan area (Vaz 2009). Packaging waste is composed of 2.45% putrescibles, 10.58% paper and cardboard, 60.8% of plastics, 3.98% of glass, 12.71% of composites, 4.98% of ferrous metals, 0.21% of non-ferrous metals, 0.02% of wood, 1.01% of textiles, 1% of fine particles, and 0.53% of others, provided by EGF (2009, personal communication, Empresa Geral do Fomento). Other default characteristics were collected from literature values like Rotter (2004); Dehoust et al. (2002), and Fricke et al. (2002). Emissions resulting from waste collection and shipping were modeled based on Borken et al. (1999); Knörr et al. (1997); Schmidt et al. (1998), and EMEP/EEA (2009).

3.2.2 Sorting plants

Since the AMARSUL is highly likely to have an automatic sorting plant in the future, the technology assessment was carried out for this reason. The packaging waste materials to be sorted are high-density polyethylene, low-density polyethylene, EPS, PET, mixed plastics, glass, composites packaging, and ferrous and non-ferrous materials. Data derived was based on processing 1 tonne of packaging waste in this recycling operation as shown in Table 4. However, manual sorting will still be employed when handling the paper/cardboard waste streams. Table 4 also shows the auxiliary material consumptions during this operation on a per tonne basis; these data are useful for the life cycle impact assessment.

Table 3 Data requirement for collection and transport waste life cycle stage

Waste collection and transport	MSW	BMW	Packaging waste	Paper/cardboard waste	Glass waste
Distance (km)	1,699,646	121,355	641,334	446,296	179,672
Diesel fuel consumption (l/100 km)	49.6	49.6	65.0	94.6	78.3
References	Gomes and Rodrigues (2010, personal communication, Moita municipality); Pinto (2010, personal communication, Almada municipality); Canta (2010, personal communication, Montijo municipality); Aleixo (2010, personal communication, Palmela municipality); Didelet (2010, personal communication, Seixal municipality); Valério (2010, personal communication, Alcochete municipality)		EGF (2009, personal communication, Empresa Geral do Fomento); Gomes (2009)		

Table 4 Consumptions and requirements of operational units

Operational units	Operational requirements	Auxiliary materials (per ton waste input in the operation)	References
Packaging MRF	Material recovery rate, 90%	Electricity (kWh), 20.92 Diesel (l), 2.01 Lube oil (8 l), 0.20 Steel (kg), 1.20	Rodrigo and Castells (2000); EGF (2009, personal communication, Empresa Geral do Fomento); (Rodrigues 2009)
Paper/cardboard MRF	Material recovery rate, 90%	Electricity (kWh), 5.35 Diesel (l), 0.64 Lube oil (l), 0.01 Steel (kg), 1.20	Rodrigo and Castells (2000); EGF (2009, personal communication, Empresa Geral do Fomento)
AD	Mechanical step Refuse—2.8% Ferrous metals recovery rate for recycling—99%	Electricity (kWh), 34.8 Diesel (l), 1.16(l) Lube oil (l), 0.12	EGF (2009, personal communication, Empresa Geral do Fomento)
	Biological process Biogas production—380 m^3/t organic waste Post-composting Decomposition rate—30% Maturation step Rejects (%)—5	Biological process Water (l)—279 Post-composting Electricity (kWh)—10 Structural material (%)—5 Maturation step Electricity (kWh), 10 Water (%)—20	Vogt et al. (2002); EGF (2009, personal communication, Empresa Geral do Fomento); APA (2009)
AD MBT	Material recovery for recycling: mainly metals, 95% Material recovery for RDF (when applied), 98 of high calorific material	Electricity (kWh), 34.8 Diesel (l), 1.16 (l) Lube oil, 0.12 (l)	EGF (2009, personal communication, Empresa Geral do Fomento)
	Biological process Biogas production—380 m^3/t organic waste Post-composting Decomposition rate—50% Maturation step Rejects (%)—10	Biological process Water (l)—279 Post-composting Electricity (kWh)—10 Structural material (%)—5 Maturation step Electricity (kWh), 10 Water (%)—20	Vogt et al. (2002); EGF (2009, personal communication, Empresa Geral do Fomento)
Aerobic MBT	Material recovery for recycling: glass, 1%; plastic, 7%; ferrous metals, 97%; non-ferrous metals: 14%	Electricity (kWh), 34.8 Diesel (l), 0.5 Lube oil (l), 0.12	EGF (2009, personal communication, Empresa Geral do Fomento);Wallmann and Fricke (2000)
	Biological step Decomposition rate—65% Maturation step Decomposition rate—20%	Electricity (kWh), 10 Diesel (l), 0.12 Water, 2% for biological step; 20% for maturation Structural material, 8.2%	Fricke and Müller (1999); EGF (2009, personal communication, Empresa Geral do Fomento) Vogt et al. (2002)
Landfill	Annual precipitation (JNS), 1,550 mm Leachate production during phase A (N24T1) 40% Leachate production during phase B (N25T1) 8% Duration phase A (PHAA) 10 years Duration phase B (PHAB) 20 years	Electricity (kWh), 0.002 Mechanical energy (kJ), 10.99 Heat energy (kJ), 1.6	Rettenberger (1996); Rettenberger and Stegmann (1997); Weber (1990); Eggels and van der Ven (1995); BUWAL (1998)

3.2.3 Anaerobic digestion

Within the AMARSUL system, it is expected to adopt a combined MBT unit, in which two separate lines will be laid out to process MSW and BMW, respectively. In regard to the BMW processing line, a small mechanical treatment process will be installed to remove unnecessary matter like metals and plastic waste destined for biological treatment. Organic waste portions delivered to the BMW unit may be decomposed in a thermophylic, dry anaerobic digestion process, resulting in a digestate. This digestate material may be sent to a post-composting unit to have the residual organic waste decomposed, producing fresh compost. To produce useful compost, a maturation phase must be arranged to produce mature compost, which can be applied for agricultural use. The requirements applied to this phase are shown in Table 4.

Emissions occurring during the anaerobic digestion mainly result from biogas burning, wastewater treatment, and gas treatment. Biogas can be used to produce electricity and heat in the process, and the amount of emissions can be modeled based on some previous work (Soyez et al. 2000; Vogt et al. 2002). Wastewater characteristics were drawn from Loll (1994, 1998) and Vogt et al. (2002) for a typical treatment process with aeration tank, reverse osmosis, sewage sludge drying through flotation, and dehydration included (EGF 2009, personal communication, Empresa Geral do Fomento). To model such a wastewater treatment plant, data from Martinho et al. (2008) and Yamada and Jung (2007) were used. The biogas treatment process may be simulated and predicted based on a biofilter, in which the average air pollutant concentration and treatment efficiency were applied with the aid of literature data (den Boer et al. 2005).

3.2.4 Anaerobic digestion MBT

The AD MBT will be located in the Seixal municipality. This unit is composed of mechanical sorting to remove recyclables and combustible fraction for RDF production, allowing the remaining fractions to be sent to the anaerobic digestion unit. Normally, the mechanical sorting process includes flail mills, trammels, magnetic separator, eddy currents separator, and ballistic separator. Sometimes manual sorting is included too to separate materials for recycling and RDF. Table 4 lists the material consumption and requirements needed to simulate the process.

After being sorted, the remaining fractions of waste with mechanical and biological recovery potential may be treated by the thermophylic, dry anaerobic digestion resulting in a digestate with several decomposed substances. The residual parts may be decomposed further through the use of an aerobic treatment process. It may lead to the production of fresh compost. After this process, fresh compost is still not mature, and it must be deposited in piles for 11 more weeks, to produce mature compost. The main parameters used to model the AD phase are listed in Table 4. The biogas produced as an integral product of the AD MBT process may be used to generate electricity. The final residuals may be used as daily cover materials at landfills. The engineering design basis applied to model the emissions in AD MBT was considered the same as those applied in anaerobic digestion of BMW.

3.2.5 Aerobic MBT

An aerobic MBT is composed of a mechanical sorting processing unit and a biological treatment processing unit, respectively. The mechanical processing unit is designed to remove the waste stream that is not relevant for the biological treatment unit. Concerning the mechanical separation, which also includes manual sorting, the materials removed for recycling are mainly ferrous and non-ferrous metals as well as some glass and plastics.

In an aerobic treatment process, the requirements applied to decompose the organic fraction of waste are presented in Table 4. From such a MBT process, the main output is the “stabilized residue”, which must be landfilled or used as the daily cover materials in landfills. In this MBT, there is no wastewater generation and contaminated air may be treated by a biofilter. The engineering design basis applied to model this biofilter was considered the same as those applied for the other similar biological treatment processes previously described.

3.2.6 Landfill

The waste stream which goes to a sanitary landfill has different sources varying from mixed MSW to residuals associated with several operational units in the MSW management system. The emissions from landfills diffuse into air, soil, and water. Typical sanitary landfills have two types of collection systems. One is for leachate collection and the other is for biogas collection. The existing one in the AMARSUL system is for the collection of biogas (i.e., methane gas) collection to produce electricity. Landfill methane gas emissions are due to biological decomposition and meteorological conditions at the local scale. The landfill module in UMBERTO was then built based on several sources in this study (Rettenberger 1996; Rettenberger and Stegmann 1997; Weber 1990; Eggels and van der Ven 1995; BUWAL 1998). The formula used to quantify the methane gas production was derived based on some literature values as adapted below (Tabasaran and Rettenberger 1987):

$$\mathrm{Ge} = 1,868^{*}\mathrm{Co}^{*}(0.014\,T + 0.28) \quad (1)$$

in which Ge is the potential methane gas production at long-term (cubic meters per tonne of waste), 1,868 is gas production rate resulting from decomposition per kilograms of organic waste [biogas per kilogram carbon] (note that [(22.4 L biogas/mol)/(12 g C/mol)=1.868 L biogas/g C]), and Co is the content of the organically degradable carbon in household waste [kilograms of Co per tonne of waste] (i.e., typical figures are 170–220 kg/tonne) Co [cubic meters of biogas per kilogram of Co]. Within the current model Co is calculated based on the C content of biologically degradable organic waste. (0.014 T+0.28) is temperature-dependent decomposition rate [in °C] (note that for household waste landfill *T* lies between 30°C and 35°C).

Air emissions due to biogas management can be attributed to direct emissions, from burning biogas and diffused emissions from landfills. Diffused emissions are linked with the arrangement of the biogas collection system during landfill operation and post-closure (phases A and B, respectively). Based on Umberto module, 25% of the biogas collected was considered as direct emissions throughout the operation and post-closure. During phase A, 30% of the biogas was considered as released emissions whereas during phase B, this number is potentially up to 70%. It is assumed that for the entire landfill life cycle around 50% of biogas from phases A and B is actually produced. Hence, in phases A and B, we have (75/100)(30/100)(50/100)=11.25% and (75/100)(70/100)(50/100)=26.25% of diffused biogas. For the amount of biogas collected, it may be estimated as (75/100)(50/100)(1–70/100) for phase A and (75/100)(50/100)(1–30/100) for phase B.

Landfill gas (LFG) energy recovery is performed using a gas turbine. The emissions from LFG burning and electricity production were calculated based on the average values collected by den Boer et al. (2005). With regard to landfill leachate, its production is also divided corresponding to phases A and B. The planning horizon is 100 years. The leachate production level depends on the annual average precipitation as well as the water content inside landfills. In the operation phase (phase A), leachate production can be estimated as between 10% and 50% of the total annual precipitation (Schwing 1999). After closure (phase B), leachate production can be as low as 5–10% of total annual precipitation. In the UMBERTO module, the default values are 40% and 8% for phases A and B, respectively (Rettenberger and Schneider 1996). Such values were applied for every type of waste. It is assumed that, based on German landfills, MSW has a residual water content of 15% by weight in which 76% may be collected as leachate (Schwing 1999). It is also assumed that leachate collection systems at landfills can collect 90% of the leachate produced. If landfilled waste has a density of 1 tonne/m^3, the land use requirement can then be determined by the ratio between the volume of waste landfilled and the soil, given that 20-m of height was applied.

3.2.7 *Products shipping*

Recyclables, compost, high calorific fraction, and RDF resulting from MSW management system have to be transported to their final destination. The shipping distance parameters are listed in Table 5. These distance parameters were obtained by using the Google map tool (Google maps 2010) and diesel consumption records collected from transportation companies based on 25 l/100 km (JMFF 2008, personal communication, José Maria Ferreira e Filhos).

3.2.8 *Auxiliary materials and recyclables*

Auxiliary materials like electricity, diesel production and burning, and lubricating oil consumption in MSW management systems were discussed by Frischknecht et al. (1996), GEMIS database (GEMIS 2001); ifeu (2009); EMEP/Corinair (2007), and EMEP/EEA (2009). In the case of lubricating oil, the data used in this study were adapted from Martinho and Pires (2008). The rest of auxiliary materials used in this study were drawn from the literature. Expansion of the system boundary due to the processing of recyclables is summarized in Table 6. The specific auxiliary materials used during recycling processes were modeled based on relevant data in Table 6.

3.3 Life cycle impact assessment

Our LCA was then carried out using the Umberto 5.5 (2009) software package with the aid of the entire LCI as described in the previous section. Following the methodology suggested by the ISO 14040–44 standard (ISO

Table 5 Distances between MSW management system and final deposition sites for products

Products transport	Distances (km)	
	Pre-processors	Recyclers/incineration[1]/agriculture[2]
Ferrous metals	241.3	521.5
Non-ferrous metals	259.3	592.2
PE	0	238.6
PET	0	210.7
EPS	0	293.0
Mixed plastics	0	524.0
Paper/cardboard	339.9	811.2
Composites	210.2	1,116.5
Glass	233.0	60.5
RDF[1]	0	45.4
Compost[2]	0	73.7

Table 6 Summary of LCI data sources for expanded systems and avoided products

Type of data	Sources of data
PET recycling, mixed plastics recycling, glass pre-processing and glass recycling	Alves (2010, personal communication, Extruplás company); ProBas (2004); APA (2009); Mata (1998)
RDF production	Fricke et al. (2003)
RDF incineration	UBA (1999); Achernbosch and Richers (1997, 1999); Schäfl (1995); Valorsul (2008)
Paper and cardboard pre-processing, composites packaging pre-processing	Rodrigo and Castells (2000)
Paper and cardboard recycling	ProBas (2004); APA (2009)
PE recycling	Arena et al. (2003)
EPS recycling	Silva (2010, personal communication, Plastimar company)
Composites recycling	Stora Enso (2008)
Ferrous metals pre-processing	Rodrigo and Castells (2000)
Ferrous recycling	ETH Zurich (2008)
Aluminum metals pre-processing	Rodrigo and Castells (2000)
Aluminum recycling	Boustead (2000)
Auxiliary materials production	APA (2009); APME (1995); Patyk and Reinhardt (1997); BUWAL (1998); Ecoinvent (2006); GEMIS (2001)
Avoided products, including fertilizers	APA (2009); BUWAL (1998); ProBas (2004); Mata (1998); ifeu (1994)

2006a, b), environmental indicators were obtained for different impact categories. The characterization factors applied to each impact category are those proposed by the CML 2000 method (Guinée et al. 2002). The impact categories studied were: abiotic depletion, acidification, eutrophication, global warming, human toxicity, and photochemical oxidation. Life cycle impact assessment can then be carried out by linking these designated impact categories with those prescribed operational efforts in Table 2.

The differential contribution of each operation unit associated with each alternative is summarized in Fig. 3. All of them are compliant with the Packaging and Landfill Directives. Finally, the ultimate environmental impact in terms of each selected life cycle impact category associated with these 18 alternatives can be independently calculated and presented by a comparative approach in Fig. 4. The following subsections discuss the pros and cons of each alternative with respect to the ultimate environmental impact associated with these designated impact categories.

4 Discussion of impact assessment

4.1 Depletion of natural resources

This impact category indicator is related to the extraction of natural resources (including energy resources) such as iron ore, crude oil, and wind energy, which are regarded as non-living materials (Guinée et al. 2002). The alternatives being assessed in our case study exhibit clear traits. It is indicative that those alternatives that have more options for resources substitution end up having better environmental performances. Those preferred alternatives include A4′, A4*, A5′, A5*, A0′, A0*, and A2′ and A2* that can produce more electricity from high calorific fraction direct burning or RDF, and biogas combustion. Naturally, alternatives with higher recycling rates due to separation at the MBT plants are favored in comparison with those that can only recycle packaging waste at the source location. Landfills with methane gas recovery leading to the generation of electricity and the avoidance of the consumption of fuel resources were favored too. The worst scenario is the base scenario which is literally the current situation.

4.2 Acidification

Acidifying pollutants have a wide variety of impacts on soil, groundwater, surface waters, biological organisms, and ecosystems and materials (buildings) (Guinée et al. 2002). Since all alternatives present the impact of acidification due to the lower emissions of NO_x, SO_2, and ammonia, alternatives A4′, A4*, A5′, A5*, A2′, A2*, A0′, and A0* particularly signify such cons due to the combustion of RDF and high calorific fractions of waste, which substitutes energy production in the power industry. On the contrary, alternatives A6 and A6* present less advantages due to more compost production given that compost application induces the release of ammonia, and consumes more energy via burning more fuel. The base scenario is the worst one again in this impact category.

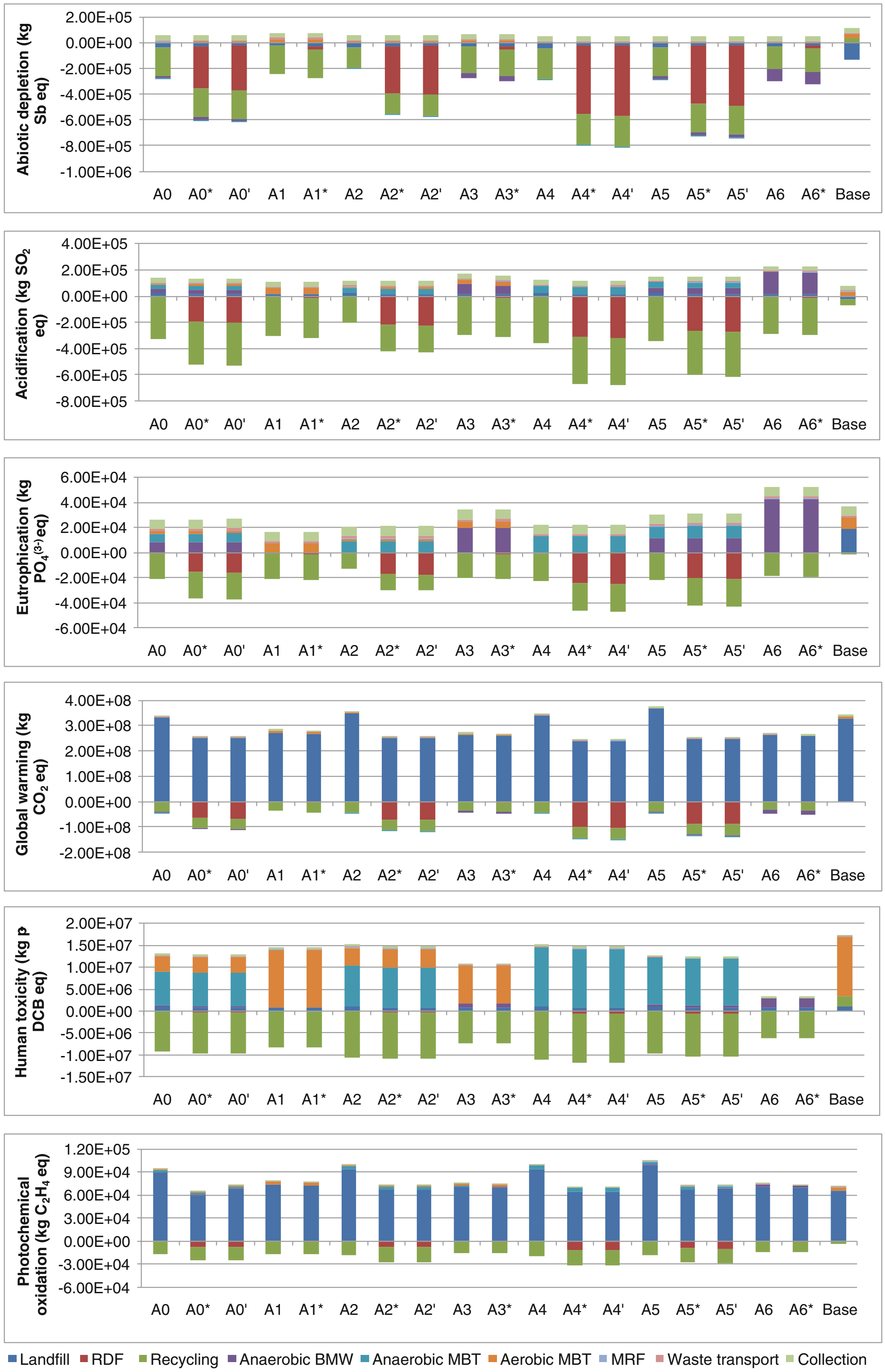

Fig. 3 Contribution made by each stage of the waste management life cycle to each impact category

4.3 Eutrophication

Eutrophication covers all potential environmental, ecological, and public health impacts due to the presence of nutrients including nitrogen and phosphorus species. Nutrient enrichment may cause an undesirable shift in species composition and surplus biomass production in both aquatic and terrestrial ecosystems (Guinée et al. 2002). The alternatives with lower release of nutrient substances include A4′, A4*, A2′, A2*, A5′, A5*, A0′, and A0*. Those alternatives were picked up because of the reduced emissions of these nitrogen substances via burning RDF

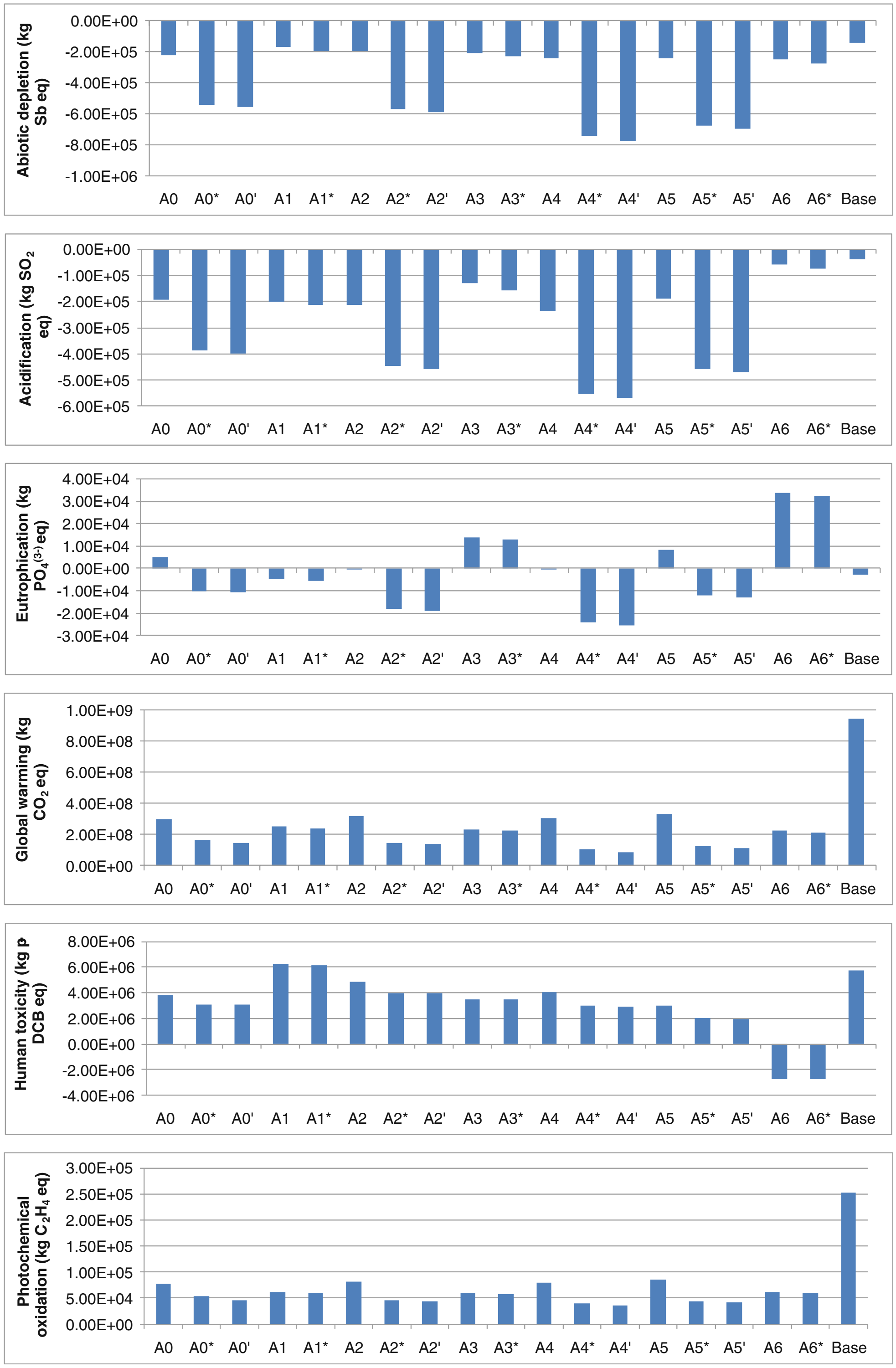

Fig. 4 Net contribution of each scenario to each impact category

and high calorific fractions of waste. However, A1 and A1* were screened out since there is no wastewater discharge during the operation of an aerobic MBT unit since the AMARSUL can reuse all wastewater effluents. This explanation can also justify the good results in the base scenario. Alternatives that consider compost production and soil amendment (A6, A6*, A3, and A3*) are penalized for the same reason as in the acidification category impact assessment.

4.4 Climate change

Climate change is defined as the impact of human emissions on the radiative forcing (i.e., heat radiation

absorption) of the atmosphere. Most of the climate relevant emissions enhance radiative forcing, causing the temperature of the Earth's surface to rise, which is referred to as the "greenhouse effect" (Guinée et al. 2002).

Recycling contributes to reduce global warming potential (GWP) substantially across all alternatives. The other important factor that is intimately linked with GWP is the amount of BMW that goes into landfills and the amount of RDF production and burning. Alternatives A4′, A4*, A5′, A5*, A2′, A2*, A0′, and A0* were selected as the best alternatives due to the production of electricity by using biogas, high calorific fractions of waste, and RDF. Besides, all these selected alternatives can redirect considerable amounts of waste streams that would otherwise be destined for landfilling leading to the generation of more GWP. This is why alternatives A4, A5, A2, and A0 were not favored as well as the base scenario.

4.5 Human toxicity

This impact category is related to the negative impacts of the toxic substances released to the environment on human health. MBT plants have higher potential of emissions of heavy metals, thereby creating negative environmental impact. The emissions from MBT plants are related to indirect emissions not only from electricity and auxiliary materials production (lube oil, diesel), but also from wastewater produced during anaerobic digestion in the MBT process. Once organic waste from source separation has a lower heavy metals content than organic waste from commingled MSW, leaching effects during decomposition are considerably higher at an MBT than the process during anaerobic digestion after source separation, which renders the alternatives A6 and A6* as the cases with lowest negative environmental impact on human health.

Because of the lack of generation of wastewater in an aerobic MBT, alternatives A1 and A1* are not favored. This can be explained by the fact that the avoided subsystems (i.e., wastewater treatment) are not enough to compensate the heavy metals being emitted from an aerobic MBT and the consequence of no electricity production using biogas. In general, other biological treatment plants in dealing with selective organic waste fractions may have considerably lower emissions of heavy metals and hydrocarbons when compared with MBT units.

4.6 Photochemical oxidation

Photochemical formation is the formation of reactive chemical compounds such as ozone in the troposphere, resulting from the reaction when sunlight interacts with some primary air pollutants. These reactive compounds may be harmful to human health and ecosystems, and may also damage crops. The relevant areas of protection are human health, man-made environment, natural environment, and natural resources (Guinée et al. 2002). The major fraction of tropospheric ozone formation occurs when nitrogen oxides (NO_x) and volatile organic compounds react triggered by sunlight. NO_x emissions are common in combustion processes. Therefore, the alternatives A4′, A4*, A5′, A5*, A2′, A2*, A0′, and A0*, which divert more waste streams for RDF production or direct burning of high calorific fraction via AD MBT, may present better environmental performance in terms of the generation of NO_x because less fossil fuels will be needed for combustion in power plants. Consequently, alternatives A4, A5, A2, A0, and base case were not favored due to the need of more electricity produced by using fossil fuels for operation.

Overall, recycling may contribute to avoid most environmental impacts. Alternatives using AD MBT burning of high calorific fractions of waste directly in incineration are classified as better solutions with a minimal difference from massive RDF production alternatives. This is due to the reduction in electricity consumption. Alternatives with high energy-from-waste potential exhibit comparative advantages over most of the counterparts such as aerobic MBT alternatives. Alternatives A6 and A6* with the inclusion of the recycling organic waste fraction are also competitive. Waste collection, transport, and sorting process present similar environmental impacts in terms of energy consumption.

4.7 Uncertainty analysis and reliability-based LCA

Most LCA practices aforementioned were based on the known or assumed data without the consideration of uncertainty. According to ISO 14040 (2006a), uncertainty analysis consists of a systematic procedure to quantify various sources of uncertainty introduced from many aspects via cumulative effects. They include the data variability, scarcity, and imprecision, as well as model or methodological uncertainty. In this study, the proposed uncertainty analysis focused on addressing both uncertainties associated with modeling assumptions and data variability.

One way to assess modeling assumption is to change the database being used to support modeling the operational units and auxiliary processes. Those databases apply different reference basis to reach the input–output balance. In this study, a database created by PE International (2010a, b) was applied to model Portuguese landfills in 2006 in which the electricity module was developed in 2002. After analyzing the possible changes of landfills and associated electricity production, Fig. 5 clearly indicates that such changes could significantly affect the LCA outcome. With these changes, alternatives A4′ and A4* are still the best options in terms of abiotic depletion, acidification, eutrophication, global warm-

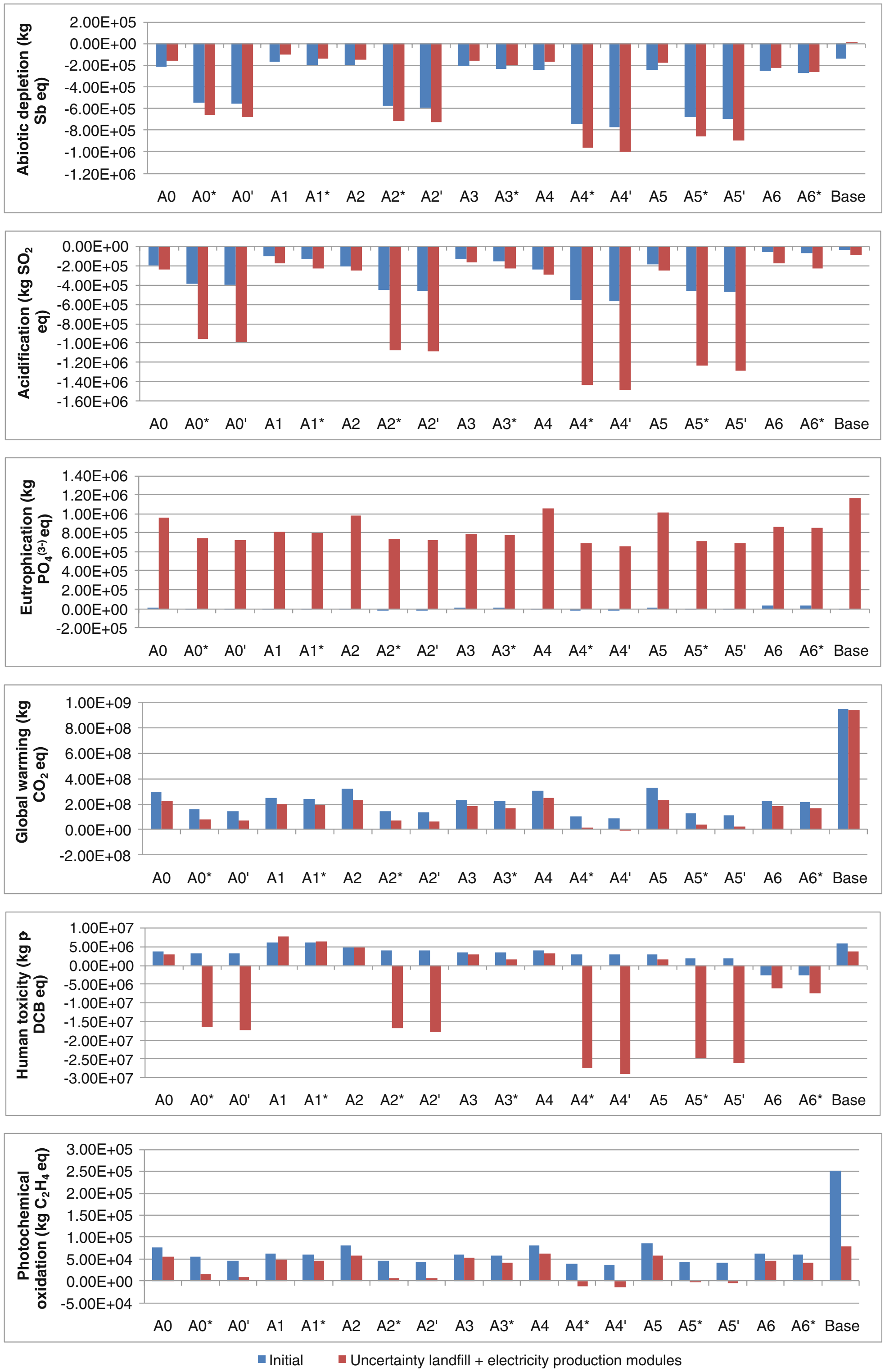

Fig. 5 Comparison of results obtained by modifying the electricity mix and landfill modules

ing, and photochemical oxidation. As for human toxicity, A4′ and A4* still appear to be the best, which is significantly different with the previous results in LCA. This difference is related to the electricity database developed by PE International, which presents higher amounts of heavy metals when compared with GEMIS/ifu data applied earlier. Similar effects are evident in other impact categories, like acidification and eutrophication.

Data variability was simulated by adjusting the levels of biogas production at these MBT plants and electricity consumption during paper/cardboard recycling by means of a Monte Carlo simulation practice based on uniform

distributions via 10,000 iterations. The range of data simulated for biogas production at the anaerobic digesters is between 320 and 450 m^3/organic dry matter. Such a range may be seen in multiple sources, including literature values (Weiland 2000) and statements given by some EGF's experts. Besides, the range of electricity consumption for paper recycling varies from 2,592 to 6,500 kJ/kg based on literature values (EIPPCB 2001) and sources from the Portuguese paper/cardboard recycling industries (APA 2009). When applying LCA to this region-based study, Figs. 6 and 7 collectively reveal that such data variations would not significantly alter the LCA results.

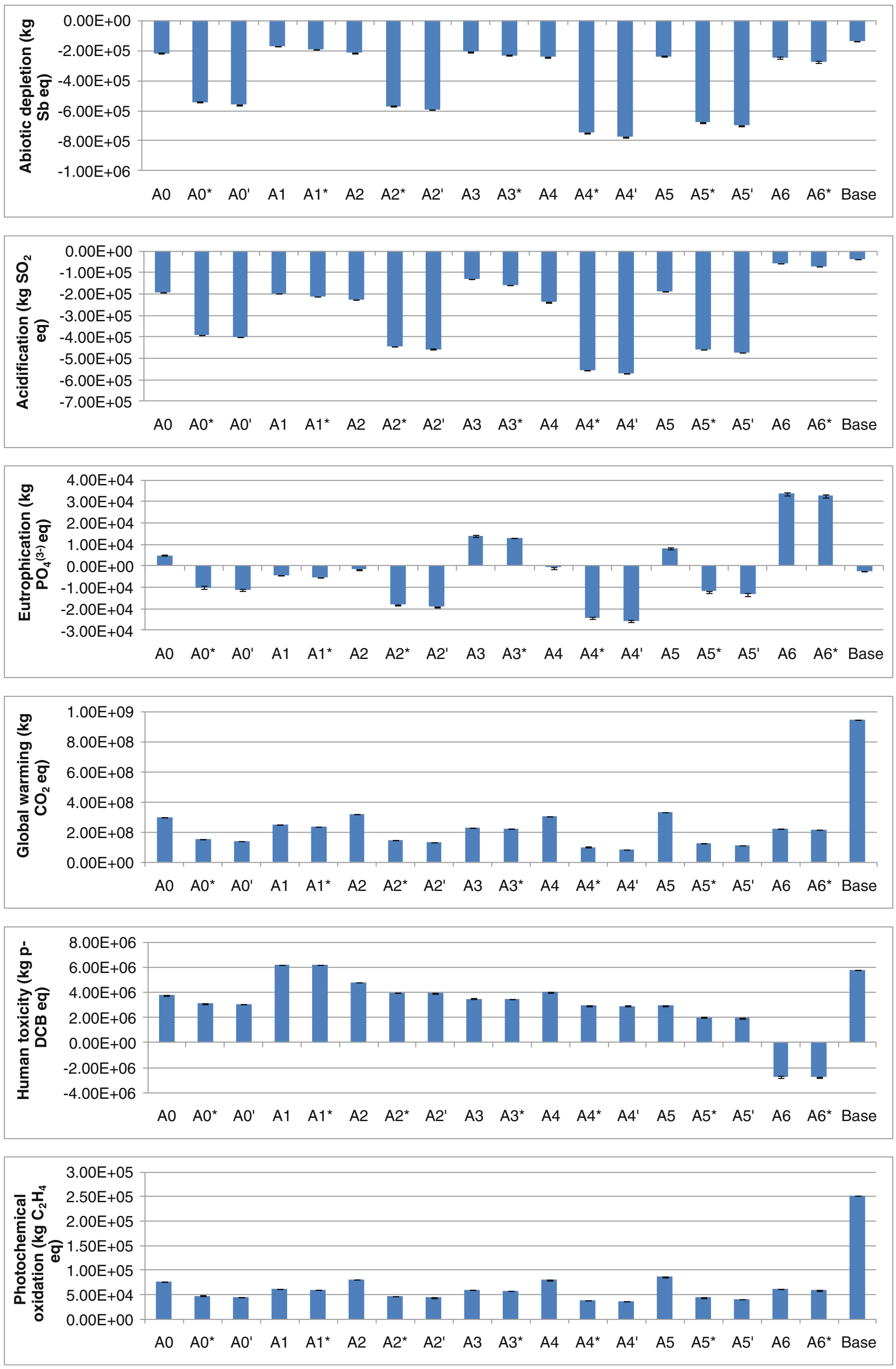

Fig. 6 Comparison of results obtained by testing biogas production in anaerobic digestion MBT processes

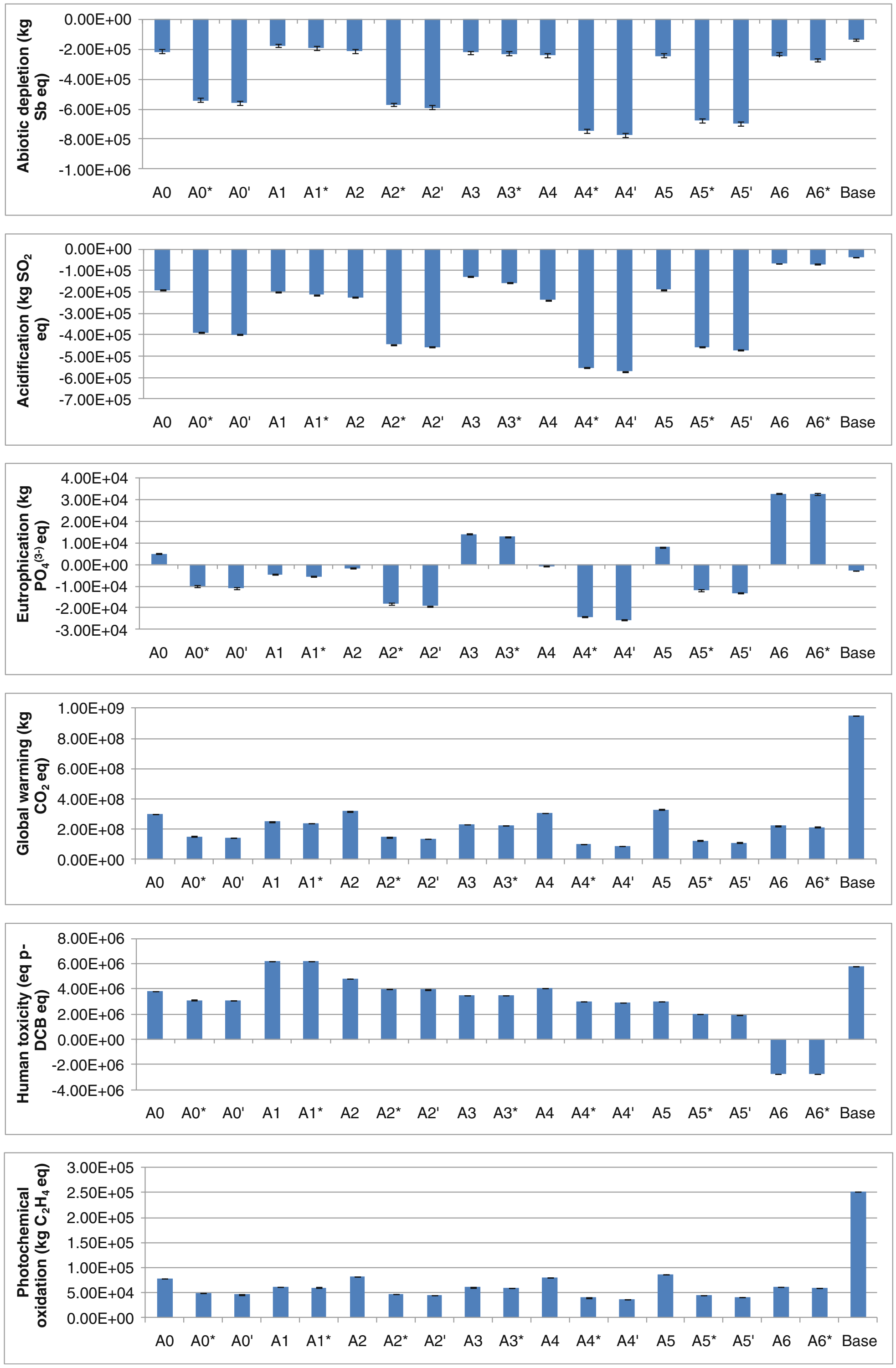

Fig. 7 Comparison of results obtained by testing electricity consumption in paper/cardboard recycling process

In addition to the concerns about data variability, there are many other dimensions of uncertainty analysis in LCA. Our reliability-based LCA is tied to the three scenarios being studied and all of them need to be tested for the robustness of the LCA outcome. The first test can be set up to answer the question: "how will the LCA results be affected by using different Portuguese electric energy LCI?" To explore this further, possible changes can be made based on the GEMIS database of Portuguese electric energy for 2030, available at ProBas (2004). The expected Portuguese electricity in 2030 has coal 30.9%, crude oil 1.18%, natural gas 35.6%, waste 4.36%, biomass 2.5%, geothermic 0.24%, hydro 24.3%, and

wind 1%. Such a reliability-based LCA study can then be characterized by an increased use of non-fossil fuels for power generation that would affect the credit of RDF production. As the result of such changes, Fig. 8 reveals that alternative A4′ is not only the best option in terms of abiotic depletion, acidification, and global warming, but also the leading one, far ahead of the rest of the preferred alternatives such as A5′, A2′, and A0′.

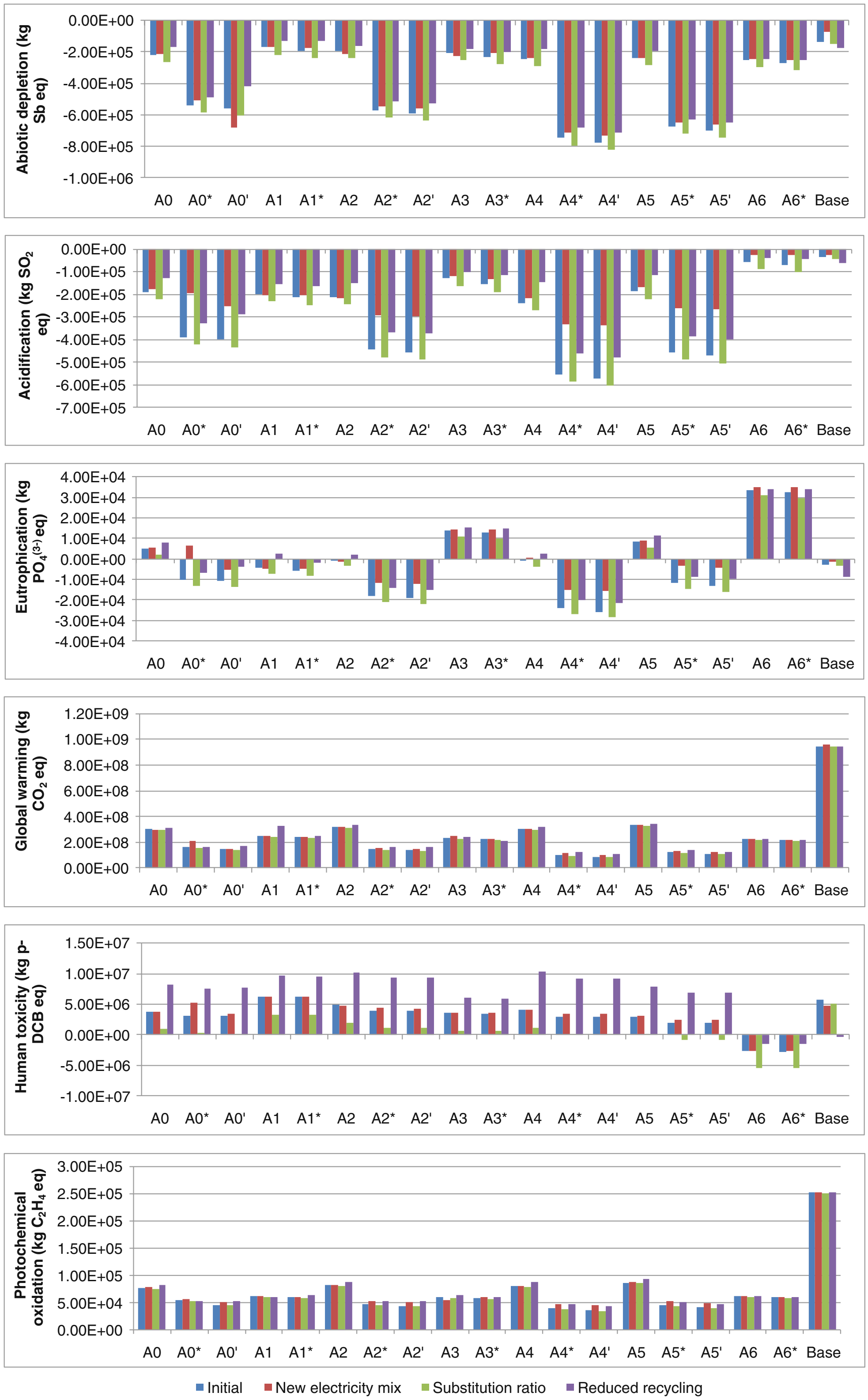

Fig. 8 Comparisons of results obtained by modifying electricity mix, substitution environmental performances

The testing of the second scenario is related to the selected substitution ratio of some recyclables as indicated in Table 2. In such a reliability-based LCA study, changes due to the inherent differences of properties between these recycled materials and virgin materials were ignored, thereby ending up with a ratio of 1:1 for testing (Bovea et al. 2010; Rigamonti et al. 2009a, b). If this is not the case, Fig. 8 also confirms that alternative A4′ is still the best option in terms of abiotic depletion, acidification, global warming, eutrophication, and photochemical oxidation. The third testing scenario deals with the situational awareness that the biological treatment units may collect a fair amount of recyclables after the installation of the automatic sorting equipment for RDF production. If we remove this premise, all the residual materials will be destined for landfilling based on the average values as shown in Table 7. Simulation results assure that both A4′ and A5′ have equivalent advantages in terms of abiotic depletion, global warming, and photochemical oxidation, both of which may be selected as the best option.

5 Conclusions

The 18 alternatives that address the current practices and possible future expansion options in 2013 were analyzed and compared with each other in the present study based on the existing LCA technologies. Initial findings clearly indicate that combination of anaerobic digestion and MBT followed by the energy recovery of the high calorific fraction of waste is an advantageous option to manage MSW, since it may not have detrimental effects in terms of abiotic resources depletion, acidification, global warming, and photochemical oxidation. Options from which the anaerobic digestion of BMW was considered can simply contribute to the reduction of the human toxicity impact. The environmental advantage of the production of RDF that is compared with direct burning of the high calorific fraction of waste in incinerators was not salient. In this case, RDF production can be justified more from a point of view of shipping advantage rather than from an energy-from-waste process itself. In fact, the LCA results show that the promotion of biological treatment is a better solution, especially when energy recovery is considered for electricity production. However, none of the alternatives studied are favored across all the impact assessment categories considered. The existence of two lines, in anaerobic digestion MBT, as biological treatment options, in which one for BMW and the other for MSW, is a positive option at least from the environmental point of view. However, the environmental impacts related to compost application have not yet been quantified fully in terms of carbon sequestration in soil and soil erosion prevention that could bring up some more positive effects and a significant environmental advantage. Hence, with reservation, we would not encourage stabilized residue applications at this juncture.

Reliability-based assessment contemplates the influence with respect to three scenarios related to electricity production, varying substitution ratio for recycling, and ignorance of recyclables that can be possibly obtained at the MBT plants. In particular, the analysis confirmed the efficacy of reliability-based LCA. The uncertainty analysis was concerned with the biogas production at a MBT plant and the electricity consumption in paper/cardboard recycling both of which are related to data assumptions. To explore the modeling assumptions, uncertainty analysis focused on the use of different input–output modules for landfill operation and electricity production. With changing data assumptions, the LCA still maintains the options previously chosen without regard to the uncertainties of concern. Yet the reliability-based assessment in dealing with modeling assumptions brings up different amounts of air pollutants in the assessment. This resulted in changes in the category impact of human toxicity, which is significantly different to the previous results in LCA.

Table 7 Reliability analysis considering different quantities of recycled materials

Recycled materials	Initial quantity (t)							Uncertainty analysis quantity (t)
	A0 group	A1 group	A2 group	A3 group	A4 group	A5 group	A6 group	
Glass	17,700	17,600	17,815	17,590	17,920	17,920	17,475	17,475
PET	1,890	1,950	1,910	1,895	1,880	1,880	1,780	1,780
EPS	200	200	200	200	200	200	200	200
PE	2,340	2,580	2,410	2,350	2,310	2,310	1,900	1,900
Mixed plastics	2,310	2,340	2,320	2,310	2,310	2,310	2,255	2,255
Composites	1,250	1,140	1,260	1,135	1,320	1,320	1,140	1,137
Ferrous metals	3,310	3,700	3,590	2,730	3,645	3,770	1,322	1,060
Non-ferrous metals	715	460	810	480	980	980	535	365
Paper/cardboard	20,340	19,650	20,410	19,650	20,755	20,755	19,655	19,650

It can thus be concluded that the three scenarios in the context of reliability-based LCA significantly contribute to the contemplation in decision making.

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